

MANURE

Mineralizable Carbon, Nitrogen, and Water-Extractable Phosphorus Release from Stockpiled and Composted Manure and Manure-Amended Soils

T. H. Dao* and M. A. Cavigelli

ABSTRACT

Dissolved N and P transfer to runoff water may increase with surface applications and shallow soil incorporation of animal manure. Information is needed regarding water-extractable nutrient release during manure decomposition to quantify that potential transfer to runoff in permanent pastures and conservation tillage systems. Release of net mineralizable C (MIN_C), net mineralizable N (MIN_N), and dissolved reactive P (DRP) was determined in stockpiled and composted cattle (*Bos taurus*) manure and manure-amended soils at 4, 20, and 35°C for 322 d at about 60% water-filled pore space. Flushes of CO₂-C exceeding 100 mg kg⁻¹ d⁻¹, inorganic N, and DRP were released rapidly from both manures when incubated alone or as soil amendments. Dissolved P release varied inversely with sorption capacity and degree of P saturation in an Aridic Paleustalf and Torrtic Paleustoll. Net mineralizable C, MIN_N, and DRP flux densities were lognormally distributed during the 322-d incubation. Results from the lognormal modeling approach suggest that incubations needed to be performed only for as long as needed to attain the 50% maximal flux density beyond the maximum to predict MIN_C, MIN_N, and DRP release flux density distributions. Significant nonlinear relationships exist between ln(cumulative CO₂-C) and inorganic N or DRP and have an inflexion point between 14 and 20 d. The nonlinearity of the C-to-N and C-to-DRP relationships indicates multiple substrate pools and supports the use of lognormal distributions to describe MIN_C, MIN_N, and DRP release from manures and manure-amended soils and to shorten laborious incubations.

LIVESTOCK PRODUCTION in the United States increasingly occurs in confined animal feeding operations (CAFOs). Large numbers of animals are gathered in a relatively small land area where huge quantities of nutrients in feeds are imported to support the operations of dairies and poultry (*Gallus gallus domesticus*), swine (*Sus scrofa domesticus*), and beef cattle production facilities (CAST, 1996). Large-scale CAFOs faced with increasing pollution potential are managing manure as a waste product (Natl. Res. Council, 1993; CAST, 1996). This results in intensive land applications of manure in the immediate vicinity of the feedlots because of the high cost of transporting manure for distances greater than about 30 km. Such management practices have resulted in nutrient-loaded soils in agricultural fields near CAFOs (Zhang et al., 2002).

Animal manure has long been used as an organic source of plant nutrients and organic matter to improve the physical and fertility conditions of agricultural lands.

Much of land-applied manure is surface-broadcast or incorporated to a shallow depth of fields under crop residue and conservation tillage management because of the need to conserve stored soil water for crop production (Unger and Parker, 1976; Dao and Nguyen, 1989; Dao, 1993; Schwartz et al., 2002). Incorporating manure using inversion or disk tillage may improve the efficiency of manure nutrient utilization by crops, but intensive tillage often eliminates the benefits associated with conservation tillage practices such as reduced soil water evaporation, erosion and runoff control, and soil C storage (Jones et al., 1985; Gilley et al., 1997; Dao, 1993, 1998; Allmaras et al., 2000). Large quantities of stockpiled cattle and poultry manure or manure mixed with bedding materials are also applied to Conservation Reserve Program grasslands and pastures (Marshall et al., 2001). As a result, manure particulates remain on the soil surface, causing elevated flow-weighted NH₄-N and DRP concentrations in runoff after manure applications (Pierson et al., 2001). Water-extractable manure nutrients may directly discharge to surface waters or build up at the soil surface and increase the potential for N and P losses and contamination of surface and ground water via surface and subsurface transport mechanisms (Liebhardt et al., 1979; Sharpley et al., 1993; James et al., 1996; Jansen et al., 2000; Zhang et al., 2002).

Information regarding decomposition and particulate nutrient mineralization in manures at the soil surface is needed. Insights into the decomposition process may be gained from the knowledge of transformations that occur during the composting of animal manure to which no additional materials (i.e., crop residues, wood chips, saw dust, etc.) have been added. The N fraction has been extensively studied to predict the N-supplying capacity of composts (Gale and Gilmour, 1986; Hadas and Portnoy, 1994; Thomsen and Olesen, 2000). Manure and manure compost MIN_N is associated with proteins and is thus strongly correlated with N released by digestion with pepsin (Castellanos and Pratt, 1981). Compositional differences, primarily in total C and N between manures and composts, result in significant differences in manure and compost MIN_N. A linear relationship was observed between MIN_N and MIN_C as percentage of added total N and C during the first 4 wk of incubation of 10 manures and composts added to a Typic

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Abbreviations: CAFO, confined animal feeding operation; CM, composted manure; DRP, dissolved reactive phosphorus; flux density_{max}, maximal flux density; MIN_C, net carbon mineralized; MIN_N, net nitrogen mineralized; PDF, probability density function; S-1, first first-order reaction; S-2, second first-order reaction; SM, stockpiled manure; WFPS, water-filled pore space.

Xerofluent (Hadas and Portnoy, 1994). A similar linear relationship was observed during the decomposition of plant residues and biosolids added to a Typic Albaqualf and a Typic Fragiudult (Gilmour et al., 1985).

The release of manure and compost P has also received much attention because of the negative impact agricultural P in runoff can have on water quality (Burkholder et al., 1992; Sharpley, 1995; Sharpley et al., 1993; Traore et al., 1999). For example, poultry manure released $74 \text{ kg ha}^{-1} \text{ NH}_4\text{-N}$ and $14 \text{ kg ha}^{-1} \text{ P}$ from a manure application equivalent to 10 Mg ha^{-1} following five simulated rainfalls (Robinson and Sharpley, 1995). The first rainfall accounted for 60% of the N and 40% of the P released during all five rainfalls. Composting manure often results in 30 to 50% reduction in mass (DeLuca and DeLuca, 1997). Phosphorus concentrations increased during composting because while C is lost by microbial respiration and N is lost by volatilization and leaching, P is conserved. The water extractability of P in composted manure (CM), however, was not significantly different than that in uncomposted manure (Dao et al., 2001).

Variation in the chemical composition and uncertainties in the predictability of nutrient release from manures and composts still hamper our ability to predict decomposition rates and pool sizes without using laborious long-term incubations. In addition, we need species-specific information on the mineralization and water extractability of manure nutrients, in particular P, to properly assess the environmental risks of surface application and shallow incorporation of manure in soil under permanent pasture or conservation tillage management. This study was conducted to (i) quantify the effects of temperature and soils on MIN-N and DRP release from stockpiled manure (SM) and CM and (ii) elucidate the relationships between manure decomposition and N and DRP release.

MATERIALS AND METHODS

Bulk samples of SM and commercially produced CM were obtained from a 25 000-head cattle feed yard located near Hereford, TX. The manure in feeding pens was scraped and stockpiled every four to six months. Twice a year, a commercial operator made a compost from the SM. Compost was made in windrows that were 1.5 m in height and 300 m in length without any additional amendment. The windrows were watered and mechanically aerated twice a week for a period of 6 wk. Bulk samples of SM and CM were crushed, sieved to pass through a screen with 6-mm openings, and stored moist at 4°C. Selected chemical characteristics of both materials are presented in Table 1.

Fifteen grams of SM and CM (dry weight basis) were weighed into crucibles that had built-in sintered glass filters. The manure samples were preleached with 250 mL of deionized water to determine the initial water-soluble N and DRP concentrations. The water content of the manure samples was then restored to about 60% of water-filled pore space (WFPS) by vacuum filtration (0.1 MPa), and the crucibles were placed in capped 1-L glass jars. Jar lids were fitted with rubber septa to allow headspace sampling. Manure and compost samples (1.2 g) were added to 30-g samples of Amarillo fine sandy loam (fine-loamy, mixed, thermic Aridic Paleustalf) and Pullman clay loam (fine, mixed, thermic Torrtic Paleustoll) (to attain a rate equivalent to 40 g kg^{-1}) and placed into the same types of crucibles and capped jars as the manure samples. The final water content of manure-amended soil samples was also brought to 60% WFPS with deionized water. The amended soil samples, SM, CM samples, and control jars (with no manure or soil) were incubated at 4, 20, and 35°C (± 0.5) in three separate temperature-controlled incubators (Revco Sci., Asheville, NC). The incubation jar headspaces were sampled twice a week for the first 3 wk and once a week thereafter. Jars were vented to the atmosphere after each headspace sampling to prevent build up of CO_2 higher than about 10% and to prevent anoxic conditions (Paul et al., 2001). Headspace CO_2 -C concentrations were measured using an infrared gas analyzer (Beckman model 865, Beckman Instruments, Fullerton, CA). Triplicate 1-mL headspace aliquots were collected with a gas-tight syringe and injected into a He carrier gas to determine CO_2 -C flux densities for the individual jars over the incubation time intervals.

Dissolved nutrients were determined from periodic leaching of the samples (i.e., 0, 2, and 6 d and 2, 3, 4, 5, 6, 8, 13, 18, 22, 26, 31, 37, and 46 wk). Seventy-five milliliters of deionized water was used to leach dissolved nutrients accumulated over the incubation time intervals. Twenty-five milliliters of a half-strength N- and P-free nutrient solution ($2.5 \text{ mmol L}^{-1} \text{ Ca}$, $1.0 \text{ mmol L}^{-1} \text{ Mg}$, $2.4 \text{ mmol L}^{-1} \text{ K}$, $2.4 \text{ }\mu\text{mol L}^{-1} \text{ B}$, $4.5 \text{ }\mu\text{mol L}^{-1} \text{ Mn}$, $0.3 \text{ }\mu\text{mol L}^{-1} \text{ Cu}$, $0.05 \text{ }\mu\text{mol L}^{-1} \text{ Mo}$, and $0.4 \text{ }\mu\text{mol L}^{-1} \text{ Zn}$) was used as a final wash of the leaching process to partially restore nutrient levels in the repeatedly leached samples. Suction equivalent to 0.1 MPa was applied to remove any excess moisture and re-establish sample water content to about 60% WFPS. The combined leachates were filtered through $0.45\text{-}\mu\text{m}$ membranes, acidified, and frozen until batch N and P spectrometric analyses were conducted using a flow-injection ion analyzer (Model Flow Solution III, Alpkem, Wilsonville, OR). Ammonium N concentrations were determined using the salicylic acid-indophenol blue method; $\text{NO}_2\text{-N}$ and $\text{NO}_3\text{-N}$ determinations were based on the Cd reduction method (Am. Public Health Assoc., 1998a) and DRP on the ascorbic acid-molybdate blue method (Am. Public Health Assoc., 1998b). Also, total organic N and P content of concentrated- H_2SO_4 digests of the manures and soils were determined spec-

Table 1. Selected characteristics of stockpiled and composted cattle manure and soils.

Material	Total N†	Total P†	Organic C‡	pH§	Electrical conductivity
		g kg^{-1}			dS m^{-1}
Stockpiled manure	16.2 ± 0.9	3.5 ± 0.4	258 ± 10.7	7.8	1.4 ± 0.6
Composted manure	13.0 ± 0.8	3.8 ± 0.4	114 ± 5.1	8.2	3.0 ± 1.3
Amarillo fine sandy loam	0.24 ± 0.02	0.18 ± 0.02	3.1 ± 0.1	7.8	0.13
+ stockpiled manure	0.61 ± 0.07	0.30 ± 0.02	—	—	—
+ composted manure	0.66 ± 0.09	0.34 ± 0.03	—	—	—
Pullman clay loam	1.60 ± 0.20	0.47 ± 0.10	9.4 ± 0.2	5.7	0.37
+ stockpiled manure	1.91 ± 0.27	0.58 ± 0.77	—	—	—
+ composted manure	1.96 ± 0.48	0.51 ± 0.12	—	—	—

† N and P means \pm standard deviation ($n = 3$) of concentrated H_2SO_4 acid digests.

‡ Dry combustion at 1350°C.

§ In 0.01M calcium chloride (2:1 solution/soil).

trometrically in triplicate 1.5-g samples (Am. Public Health Assoc., 1998a, 1998b).

To determine $\text{PO}_4\text{-P}$ sorption capacity of the two soils, triplicate 1.5-g samples were weighed into polycarbonate test tubes that hold 15 mL of standards solutions of $\text{Na}_3\text{PO}_4 \cdot 12\text{H}_2\text{O}$ containing 0, 0.05, 0.1, 0.2, 0.5, 1, 1.5, and 3 mmol L^{-1} P. The soil solution mixtures and triplicate tubes containing P solutions alone were agitated on an end-over-end shaker for 16 h at room temperature. After centrifugation at $10\,000 \times g$ for 20 min, the supernatant solutions were filtered through 0.45- μm membranes, and P concentrations were determined as previously described. The amount of P sorbed was calculated as the difference between the amounts of P in the control standard solutions and those in the equilibrium solutions at the end of the equilibration period. Phosphorus sorption data were fitted to the Langmuir model

$$C_e/x = C_e/b + 1/K_b$$

where C_e = equilibrium solution-phase P concentration (mmol L^{-1}), x = equilibrium sorbed-phase concentration (mmol kg^{-1} soil), b = sorption maximum (mmol kg^{-1} soil), and K_b = constant related to bonding energy (L mmol^{-1}).

Within a temperature setting, the experimental units were arranged in a factorial design. Manure types (2) and soils (2) were replicated three times and arranged in a randomized complete block pattern in each incubator. Differences in manure and soil treatment main effects and interactions were detected using analysis of variance and the Duncan multiple range test at the 0.05 level of probability using the Statistical Analysis System (SAS Inst., 1989). Kinetic analyses were made to derive $\text{CO}_2\text{-C}$ fluxes, inorganic N, and DRP release parameters and flux density distribution parameters using numerical curve-fitting methods (TableCurve 2D, SPSS, Chicago, IL).¹ Least-squares optimization procedures were used to assess convergence and goodness of fit.

RESULTS AND DISCUSSION

Evolution of Carbon Dioxide Carbon

a. Flux Densities

Flux densities of $\text{CO}_2\text{-C}$ were indicative of the chemical stability of the C pools in the two types of manure and amended soils on rewetting. Carbon dioxide flushes were highest during the first days of incubation and, at 35°C, remained high at flux densities exceeding $100 \text{ mg C kg}^{-1} \text{ d}^{-1}$ for about the first 30 and 100 d for CM and SM, respectively (Fig. 1). Rates of organic residue decomposition are typically described by first-order reaction rate equations for cumulative MIN_C (Gilmour et al., 1985; Hadas and Portnoy, 1994). The distribution of $\text{CO}_2\text{-C}$ flux densities with time is less frequently characterized, and when it is described, it is usually represented by exponential decay functions (Paul et al., 1997, 2001; Robertson et al., 1999). We observed that the $\text{CO}_2\text{-C}$ flux density distributions fitted the lognormal probability density function (PDF) for the 20 and 35°C data (Fig. 1A and 1B). This result is consistent with our knowledge that a soil or other microbially active medium's capacity to oxidize organic C to CO_2 is a function of microbial population dynamics and growth. We also know that microbial growth and decay are logarithmic in na-

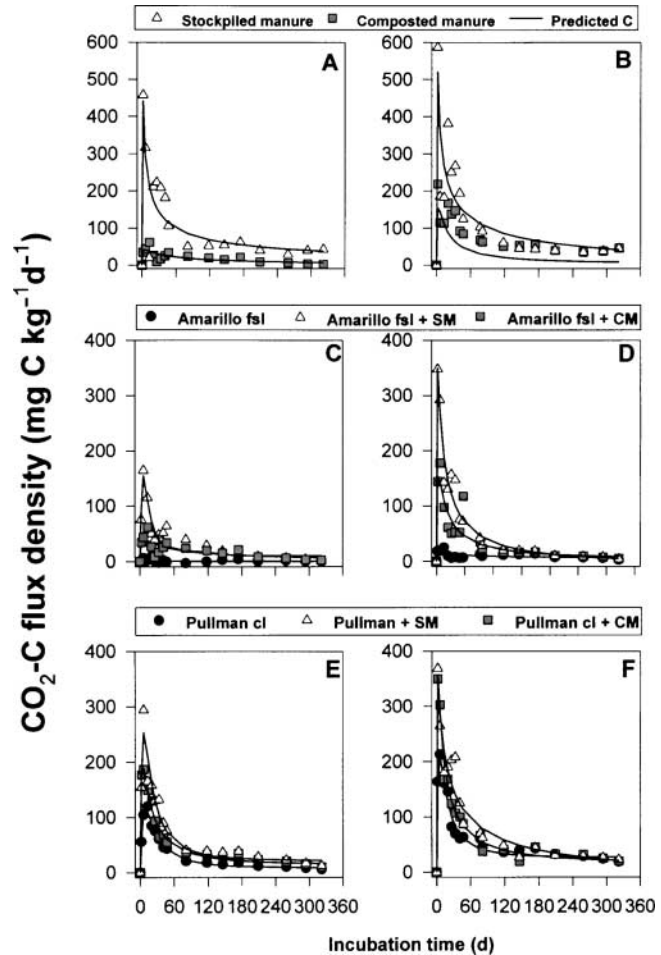


Fig. 1. Evolution of $\text{CO}_2\text{-C}$ from (A and B) stockpiled (SM) and composted (CM) manure, (C and D) Amarillo fine sandy loam (fsl), and (E and F) Pullman clay loam (cl) amended with the manures at the rate of 40 g kg^{-1} during a 322-d incubation at (A, C, and E) 20°C and (B, D, and F) 35°C. Solid lines represent the predicted lognormal probability density distribution of $\text{CO}_2\text{-C}$ flux densities.

ture. In fact, our data closely agree with those presented for microbial biomass dynamics during long-term soil incubations by Nicolardot et al. (1994). Our result also suggests that the distribution of manure MIN_C flux densities can be characterized simply from the knowledge of three parameters derived from the lognormal PDF: (i) the flux density_{max}, (ii) the time at which flux density_{max} occurs, and (iii) the width of the PDF at one-half flux density_{max}. The flux probability density distribution can be represented as follows:

$$f(x) = \frac{1}{\sqrt{2\pi}\sigma x} \exp\left[-\frac{(\ln x - \mu)^2}{2\sigma^2}\right] \quad (x > 0)$$

where μ = mean of $\ln x$ and σ = standard deviation of $\ln x$; or

$$y(x) = a \exp\left[-\frac{1}{2}\left(\frac{\ln\left(\frac{x}{b}\right)}{c}\right)^2\right] \quad [1]$$

where a = flux density_{max}, b = time (x) at flux density_{max}, (with $b \neq 0$), c is a parameter associated with the area

¹ The mention of trade or manufacturer names is made for information only and does not imply an endorsement, recommendation, or exclusion by the USDA Agricultural Research Service.

under the PDF curve, and the width of the distribution at one-half flux density_{max} =

$$b \exp[c \times (2 \ln 2)^{1/2}] - b \exp[-c \times (2 \ln 2)^{1/2}], (c > 0) \quad [2]$$

In other words, the times (x) corresponding to each of the two one-half flux density_{max}'s are $b \exp[-c \times (2 \ln 2)^{1/2}]$ and $b \exp[c \times (2 \ln 2)^{1/2}]$.

This observation implies that an extended incubation leaching of 46 wk may not be necessary to estimate manure MIN_C. Incubation times necessary to attain the one-half flux density_{max} beyond the flux density_{max} should be sufficient to derive the coefficients necessary to describe the C flux density distribution and the cumulative distribution functions (Wolfram, 1996). At 35°C, for example, that time was 13 and 26 d for SM and CM, respectively, and it ranged from 13 to 20 d and 18 to 26 d for SM- and CM-amended soils, respectively (Fig. 1). The time necessary for MIN_C to reach flux density_{max} and the magnitude of flux density_{max} reflected the effects of temperature and chemical stability of C substrates on manure decomposition. Flux density_{max} was lower in CM than SM at all incubation temperatures because CM lost most of the active C substrates during the original composting process (Fig. 1 and 2). Ambient temperature strongly influenced MIN_C, with highest C flux densities occurring at 35°C and lowest flux densities occurring at 4°C.

To illustrate the predictive approach of abbreviating incubation periods and estimating C and nutrient pool

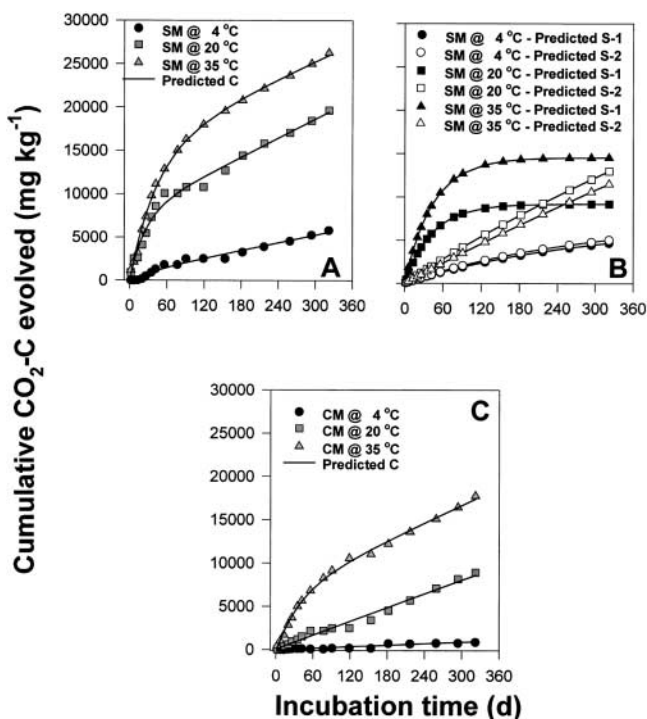


Fig. 2. Cumulative CO₂-C evolutions from (A and C) stockpiled (SM) and composted (CM) manure and predicted component reactions of the cumulative C evolved during a 322-d incubation at 4, 20, and 35°C. Solid lines represent the predicted cumulative release via the first first-order reaction (S-1; filled symbols) and the second first-order reaction (S-2; open symbols).

sizes and turnover rates from lognormal flux density functions, we derived the three coefficients a , b (Eq. [1]), and c (Eq. [2]) for CO₂-C flux density distributions for SM, SM-amended Pullman soil, and SM-amended Amarillo soil at 35°C. We then calculated the respective cumulative MIN_C distributions and contrasted the predicted results against observed cumulative MIN_C for these cases (Fig. 3). Regression analysis showed that there was no difference between observed and predicted cumulative MIN_C for SM, SM-amended Amarillo, and SM-amended Pullman soil ($P < 0.001$).

The distribution of CO₂-C flux densities from both unamended and manure-amended soils also fitted the lognormal PDF (Fig. 1C–1F), and cumulative MIN_C followed first-order reaction kinetics (Fig. 2). Adding both manures to soils significantly increased CO₂-C flux densities, compared with unamended soils, and SM-amended soils released more CO₂-C than CM-amended soils. The Pullman soil had a greater capacity to mineralize organic C substrates to CO₂ (high flux density_{max} and short lag time until flux density_{max} was reached) than did the Amarillo soil. These results are consistent with the Pullman clay loam having a higher organic carbon content (Table 1).

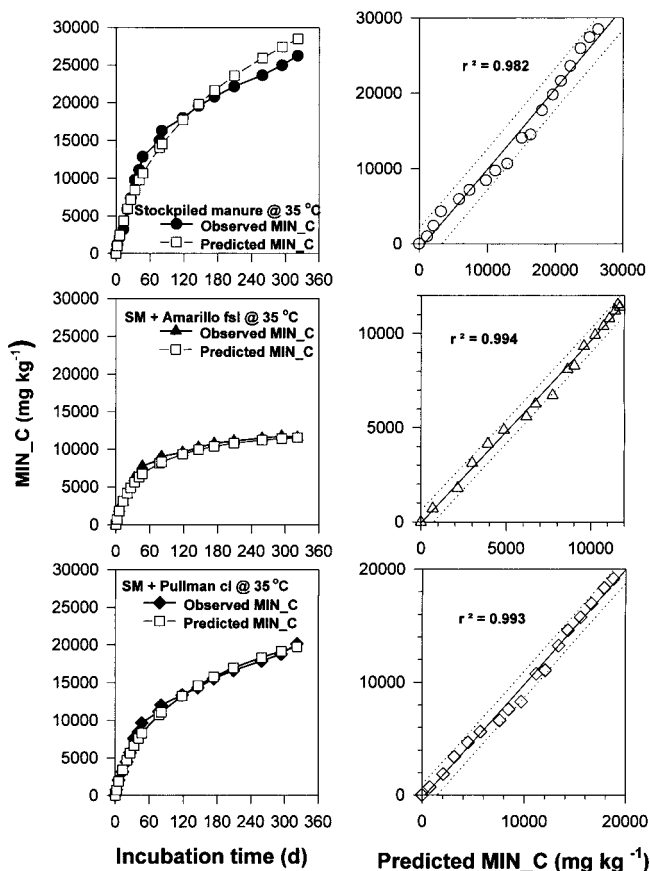


Fig. 3. Observed and calculated cumulative CO₂-C using the predicted parameters of the lognormal probability density functions during a 322-d incubation of stockpiled manure (SM) and manure-amended soils at 35°C. MIN_C, net C mineralized; fsl, fine sandy loam; cl, clay loam.

b. Cumulative CO₂-C Fluxes

Although the cumulative distribution function for the lognormal density function could be used, we fitted the cumulative CO₂-C fluxes from SM to a model consisting of two independent first-order reactions

$$y(x) = a \times (1 - e^{-bx}) + c \times (1 - e^{-dx}) \quad [3]$$

where y = cumulative CO₂-C (mg kg⁻¹), x = number of days of incubation (d), a and c = maximum accumulation concentrations (mg kg⁻¹), and b and d = CO₂-C production flux coefficient (d⁻¹) (Gilmour et al., 1985; Paul et al., 2001) (Fig. 2A and 2B). We refer to the first and second independent first-order reactions in Eq. [3] as the S-1 and S-2 reactions, respectively.

Temperature exerted a more profound influence on S-1 than on S-2 reactions. The two manure-C substrate pools were mineralized at approximately identical fluxes at 4°C (Fig. 2A), but at 20 and 35°C, the S-1 reaction contributed two and three times, respectively, the amount of CO₂-C produced at 4°C. Flux density averaged 0.22 and 0.23 d⁻¹ at 20 and 35°C, respectively. The S-1 reaction fluxes were sustained for a longer time at 35 than 20°C, resulting in a higher maximum accumulation. The CO₂-C produced via the S-2 reaction, however, reached an approximately identical maximum accumulation at 20 and 35°C (12.9 and 11.5 g kg⁻¹ C). Together, the active and intermediate C pools represent 2.2, 8.3, and 10.2% of SM total C at 4, 20, and 35°C, respectively. While CM-C is composed of complex and biologically stable C substrates, appreciable decomposition to CO₂-C occurred at the higher temperatures (Fig. 1B and 2C). It is likely that this S-1 flux represents decomposition of fine particulate organic matter. A single first-order kinetic equation best described MIN_C for CM. Accumulation maxima (Fig. 2C) and CO₂-C production flux densities (8.1×10^{-5} , 9.1×10^{-5} , and 5.5×10^{-3} d⁻¹ at 4, 20, and 35°C, respectively) were similar to those of the S-2 reaction for the SM (3.5×10^{-3} , 1.2×10^{-3} , and 0.7×10^{-3} d⁻¹), suggesting that most C substrates in CM were of intermediate stability (Robertson et al., 1999). At 4, 20, and 35°C, the cumulative CO₂-C production flux corresponded to 0.9, 8.2, and 15.6% of CM total C (CM total C was about half of SM total C), respectively.

Mineralizable Nitrogen

There were large releases of NH₄ or MIN_N that subsided to background concentrations after the first 12 to 20 d of incubation of SM and CM, with flux density_{max} occurring between 1 and 2 d (Fig. 4). Given the high active C content and oxygen demand during this period, no net NO₃-N accumulated in the manures until Day 30 when a modest release was observed in CM and in CM only (Fig. 5B and 5E). Manure additions increased MIN_N in amended soils in proportion to the amount of manure N added, i.e., NH₄-N flux densities in amended soils were about 8% of those for manures alone. Also, the Pullman soil released less NH₄-N than the Amarillo soil, which is consistent with the Pullman soil having a higher cation exchange capacity (27.5 cmol kg⁻¹) than

the Amarillo soil (7.4 cmol kg⁻¹) (R.C. Schwartz, personal communication, 2002).

Cumulative MIN_N over the 322-d period was strongly correlated to cumulative CO₂-C evolution (Fig. 5), as has been found for animal manure, plant residues, bio-solids, and residue-amended soils (Castellanos and Pratt, 1981; Gilmour et al., 1985; Gale and Gilmour, 1986). However, the relationship between cumulative MIN_C and cumulative MIN_N in our study was nonlinear in most of our manure and soil combinations. The 0- to 20- and 20- to 322-d periods showed distinct linear relationships between cumulative MIN_N and ln(cumulative MIN_C). Unlike the estimation of soil MIN_C and MIN_N from short incubations of 1 d or 3 d after rewetting dried soils (Franzluebbers et al., 1996, 2000), our results suggest that measurements of respiration in rewetted manures or manure-amended soils would have to be performed for periods longer than 20 d to get a complete picture of the active and intermediate N pools and mineralization flux densities, unless the predictive approach that we are proposing is followed. Then, characterization of N release from manures would reduce to the determination of (i) flux density_{max}, (ii) the time at which flux density_{max} occurs, and (iii) the width of the PDF at the one-half flux density_{max}.

Dissolved Reactive Phosphorus

Dissolved reactive P was released rapidly from SM and CM, and flux densities of DRP were also lognormally distributed (Fig. 4). The flux density_{max} occurred at about 1.0 to 1.8 d for SM and CM, respectively. Therefore, large and rapid releases of DRP by SM and CM potentially can be detrimental to surface water quality when provisions for P sinks are not readily available. These findings also provide additional evidence that manure additives are needed to reduce water solubility of P in untreated manure and mitigate detrimental impacts on water resources (Dao, 1999; Dao et al., 2001; Dao and Daniel, 2002).

Equation [3] described the cumulative release. For SM, DRP accumulation maxima ranged from 61 to 168 mg kg⁻¹ at flux densities ranging from 0.32 to 0.61 d⁻¹ at 4 to 35°C. The S-2 reaction contributed an additional 6.0 to 20.8 mg kg⁻¹ at flux densities between 0.01 and 0.03 d⁻¹. Overall, the cumulative DRP release from SM increased with temperature, being 24 (±0.3), 35 (±0.4), and 49 (±0.2) g kg⁻¹ SM total P at 4, 20, and 35°C, respectively. In CM, DRP accumulation maxima ranged from 23.4 to 31.7 mg kg⁻¹ at flux densities ranging from 0.44 to 0.81 d⁻¹. The S-2 reaction added another 44% of the total DRP release or 13.5 (±2.7) mg kg⁻¹ at flux densities averaging 0.031 (±0.003) d⁻¹.

Cumulative DRP release from CM [48.9 (±4.1), 51.1 (±4.1), and 33.7 (±2.2) mg kg⁻¹] was lower than for SM, which was equivalent to 12 (±0.1) g kg⁻¹ total P of the CM, at 4, 20, and 35°C, respectively. The accumulation of DRP decreased at 35°C probably because the compost had soil mixed with it that contributed sorptive capacity. Its behavior at higher temperatures will be addressed in a later section. Although the SM and CM

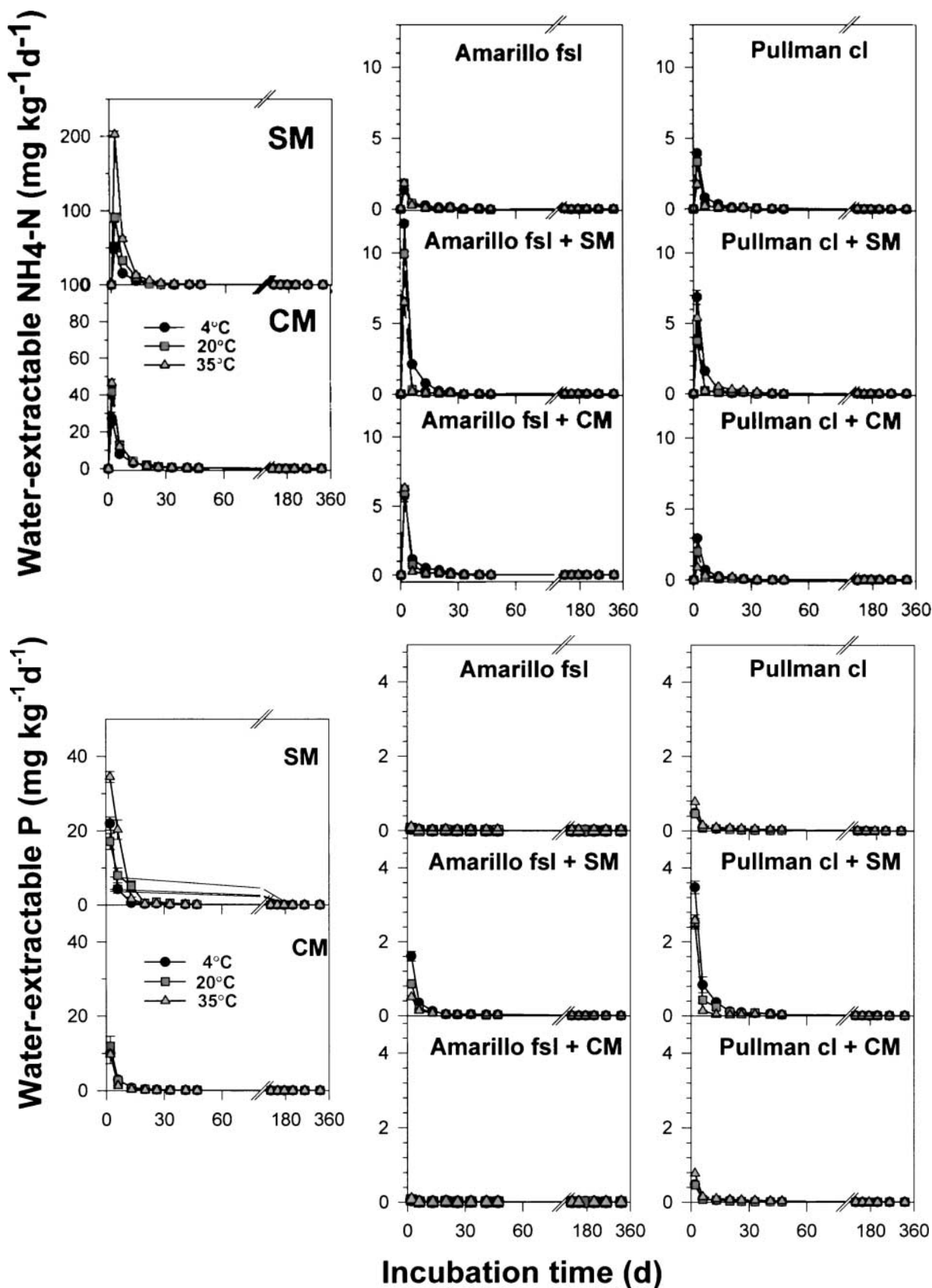


Fig. 4. Release of water-extractable $\text{NH}_4\text{-N}$ and dissolved reactive P (DRP) from stockpiled (SM) and composted (CM) manures, Amarillo fine sandy loam (fsl), Pullman clay loam (cl), and manure-amended soils during a 322-d incubation at 4, 20, and 35°C.

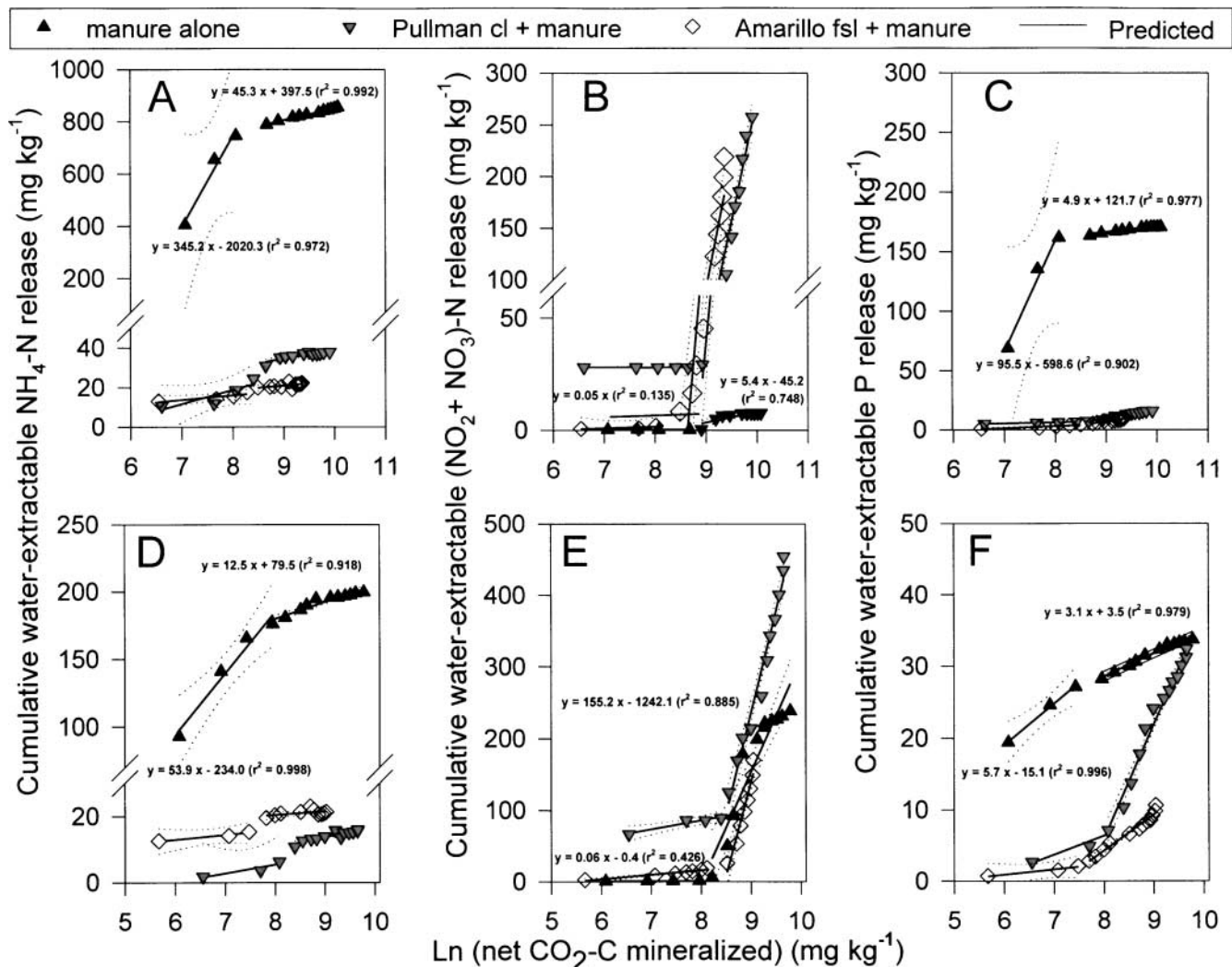


Fig. 5. Binary relationships between cumulative inorganic N and cumulative CO₂-C evolved and between dissolved reactive P (DRP) released and cumulative CO₂-C evolved from (A, B, and C) stockpiled and (D, E, and F) composted manures and manure-amended Amarillo fine sandy loam (fsl) and Pullman clay loam (cl) during a 322-d incubation at 35°C. Broken lines represent curves' 95% confidence limits.

came from the same feedlot, it appears that there are differences in the composition and storage time (age) of the manure stockpiles used by the commercial composter. Others have also observed differences in relative concentrations of DRP released from noncomposted and CM dairy manure (Sharpley and Moyer, 2000). We postulated that the P that is released quickly via the S-1 reaction is derived from a pool of dietary inorganic P excreted in manure or that had accumulated via dephosphorylation of organic P during the stockpiled period. We also suggest that the DRP release via the S-2 reaction depends largely on organic P mineralization and desorption-mediated processes.

Cumulative DRP release from the Pullman soil was higher than that from the Amarillo soil (Fig. 4). Langmuir adsorption isotherms showed that the Amarillo soil has a PO₄-P sorption potential and a binding energy, K_b , that are more than four times greater than for the Pullman soil (Table 2). The unamended Amarillo soil has a lower total P content and lower P saturation, defined as the ratio of the amount of sorbed P to the P sorption capacity, compared with the Pullman soil

(Tables 1 and 2). Thus, net DRP cumulative fluxes were greater in Pullman than Amarillo soil when amended with SM, reaching maximal accumulations of 3.1 to 7.4 mg P kg⁻¹ and 1.9 to 14.5 mg P kg⁻¹ in the SM-amended Amarillo and Pullman soils, respectively. Adsorption-desorption onto soil modified DRP release. Net DRP released was a small proportion of the total added P, that is, 1.2 and 4.9% for CM and SM, respectively, compared with <4% in a study of poultry manure P release from Delaware soils (Mozaffari and Sims, 1996).

Table 2. Phosphate P sorption isotherms† for Amarillo fine sandy loam and Pullman clay loam at 20°C.

Soil	Sorption maximum	95% confidence limits	K_b	r^2
	mmol kg ⁻¹		L mmol ⁻¹	
Amarillo fine sandy loam	15.0	14.2 and 15.7	29.6	0.987
Pullman clay loam	3.4	3.1 and 3.8	6.4	0.941

† Langmuir sorption model, $C_e/x = C_e/b + 1/K_b$, where C_e = equilibrium solution-phase P concentration (mmol L⁻¹), x = equilibrium sorbed-phase concentration (mmol kg⁻¹ soil), b = sorption maximum (mmol kg⁻¹ soil), and K_b = constant related to bonding energy (L mmol⁻¹).

Dissolved reactive P release was lower at 20 and 35°C than 4°C for amended soils (Fig. 4). Microorganisms assimilated the released DRP as microbial activity was very high. The latter condition was indicated by the high CO₂-C fluxes from manure-amended soils (Fig. 1). Dissolved reactive P is readily assimilated in biological systems and is not expected to accumulate in such a soil–manure medium. On the other hand, minimal biological immobilization of DRP occurs at 4°C when microorganisms are relatively inactive. This observation suggests that environmental risks of manure P in field soils may be greatest during cool months of the year when biological immobilization and plant uptake are low.

In manures and amended soils, cumulative DRP released over the 322-d period also was significantly correlated to cumulative CO₂-C evolution (Fig. 5). As with MIN_N, two periods, from 0 to 20 and 20 to 322 d, showed distinct linear relationships between released DRP and ln(cumulative MIN_C). After the first approximately 20-d period, coefficients of determination of these binary relationships were as high as 0.9 ($P < 0.001$) (Fig. 5). The results suggest that DRP release is still a function of manure decomposition. As with N mineralization–immobilization turnover in soils, DRP correlation to C is based on microbial energetic needs. Unlike N mineralization–immobilization, however, DRP release is also affected by precipitation–dissolution–sorption equilibria.

Implications for Manure Management in Grasslands and Conservation Tillage Systems

Our experimental results show that manure decomposed rapidly; fluxes of manure MIN_C were large early in the incubation periods, and the CO₂-C losses increase the potential for elevating atmospheric concentrations of a gas that contributes to the greenhouse effect. Inorganic N and DRP were released rapidly from both manures when incubated alone or as soil amendments. Manure DRP flushes should also be expected following periods of cool temperatures, in contrast to organic N mineralization. Therefore, applying manure infrequently at high rates or applying manure frequently at low rates that cumulatively exceed the systems' ability to assimilate manure nutrients would increase the detrimental effects of the rapid N and P release. It appears that for surface-applied manures, it is advisable to apply these materials as close to periods of plant uptake as possible to make best use of readily available nutrients and to reduce potential off-farm losses. Alternatively, N and P excretion by livestock should be reduced or manure additives used to reduce excessive nutrient levels in manures and ultimately reduce their dispersion in the environment. The N and DRP fractions that are not biologically fixed are subject to off-site transport. Therefore, surface applications or shallow soil incorporation of manures on such agronomic production fields require additional structural practices to reduce runoff water velocity and promote infiltration, in-field residence time for dissolved nutrients, and particulate redeposition (Zhang et al., 2002). Although MIN_N and released DRP

represent a small proportion of manure total N and P, repeated manure applications could result in considerable accumulation of organic materials of intermediate stability in soil that will continue to contribute dissolved N and DRP to the soil–manure–water system.

In summary, our ability to reliably and consistently determine release of N and DRP from specific manures as a function of environmental factors can be improved to accurately determine potential N and P transfer to runoff and/or subsurface transport. The lognormal descriptive approach to predict cumulative fluxes and pool sizes of environmentally sensitive manure nutrients enhances our ability to predict manure N and DRP release and transfer to soils. This tool can be used to partition manure mineralizable C and water-extractable N and DRP pools into active and intermediate pools to develop discharge mitigation strategies on grasslands and croplands under conservation tillage management.

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